Achieving Economically Feasible Drinking Water Regulation

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1. Introduction

Drinking water has been comprehensively regulated in the United States since the passage of the Federal Safe Drinking Water Act of 1974 (SDWA 1974). Like other Federal environmental statutes, SDWA 1974 assigned a dominant regulatory role to the USEPA (USEPA) in promulgating National Primary Drinking Water Regulations (NPDWRs), usually by setting uniform Maximum Contaminant Levels (MCLs), and required the States to develop programs to implement Federal standards. States that developed such programs could become *primacy* states authorized to enforce Federal law; States that did not were subject to direct regulation by USEPA. Most drinking water was (and remains) supplied by intrastate utilities relying on intrastate sources, so the logic for Federal preemption is not clear cut. State drinking water statutes mimic the Federal SDWA, in some cases establishing a greater role for economic feasibility. For example, California drinking water standards are explicitly required to be both technologically and economically feasible (Calif. HSC § 116365(b)).

SDWA 1974 sought to achieve three goals: (1) establish national standards binding on all States and localities that are as close to zero risk as technologically possible, (2) ensure that these standards are both technologically and economically feasible, and (3) accommodate significant differences in cost among water systems of different sizes relying on different sources. These three goals were in tension if not conflict from the start. Stringent national standards could provide robust public health protection only by sacrificing economic feasibility. Standards that were economically feasible for very large metropolitan water systems would not be economically feasible for the vast majority of public water systems, which were (and still are) small. Accommodating small systems' diseconomies of scale requires sacrificing stringency.

USEPA implemented SDWA 1974 by emphasizing stringency over economic feasibility and the accommodation of small systems (Schnare 1998). Congress amended SDWA in 1986 (SDWA 1986) reiterating Congress's preference for regulatory stringency. But Congress changed direction in SDWA 1996, formally directing USEPA to apply benefit-cost principles positively and normatively. This included identifying and analyzing multiple regulatory alternatives, taking account of incremental effects, and estimating risks and benefits "in accordance with sound and objective scientific practices" (§ 1412(b)(3)). Also for the first time, Congress explicitly authorized USEPA to refrain from promulgating standards whose benefits did not justify their costs (§ 1412(b)(6)(A)).

SDWA does not define *economically feasible* or any of its analogues, however. In this paper economic feasibility is defined as the condition in which benefits exceed costs, where both are typically calculated at the margin (i.e., marginal net benefits are positive). This definition is objective and grounded in economic principles. It also is intuitively appealing to non-economists, who may struggle to understand how a consumption decision ever could be economically "feasible" if it made them worse off – a necessary outcome if costs exceed benefits. Any scenario in which benefits are ignored cannot be economically feasible. Thus, *economic feasibility* explicitly excludes *affordability.*

2. Affordability

Since 1974, USEPA managers preferred to ensure that NPDWRs are *affordable* rather than economically feasible (Schnare 1998). Unlike *economic feasibility*, the term *affordability* has no objective meaning; it is not grounded in economic principles; and its intuitive appeal

among non-economists depends on its interpretation.[1] In SDWA 1996, Congress ratified *affordability* as a desirable policy goal but did not define it. Moreover, though presumably aware of USEPA's similar pre-1996 definition of affordability in the context of water quality (not drinking water) (USEPA 1995), Congress also did not necessarily endorse it. Rather, Congress directed the Agency to "publish information to assist the States in developing affordability criteria" (SDWA 1996, § 1415(e)(7)(B)). Congress also did not authorize USEPA to promulgate Federal affordability standards. Subsequently, *affordability* has become a blend of policy, guidance, and implicit regulation that is appropriately characterized as an Agency *doctrine* – "a codification of beliefs or a body of teachings or instructions, taught principles or positions, as the essence of teachings in a given branch of knowledge or in a belief system" (Wikipedia 2018).

The guidance describing USEPA's post-SDWA 1996 affordability doctrine (USEPA 1998a, b) is simple in some respects but complex in others. Its simplest features are a reliance on median household income (MHI); the choice of MHI domain (national); and the establishment of 2.5% of national MHI as the upper-bound threshold for household expenditures on drinking water deemed to be *affordable*. Multiple dimensions of variability, including regions, systems within a size category, and households served by individual systems, are not taken into account. As USEPA stated in 2006:

> Treatment technology costs are presumed affordable to the typical household if they do not cause median water bills to exceed an affordability threshold of 2.5 percent of MHI. This approach assumes that affordability to the median household in a system size category can serve as an adequate measure for the

[1] A policy is both affordable and economically feasible if and only if benefits exceed costs.

> affordability of technologies to the size category as a whole (USEPA 2006c, p. 10673).

Note that the affordability doctrine is strictly cost-based; it does not account for the benefits realized by drinking water regulation. Further, the doctrine provides only limited relief from economically infeasible NDPWRs, under limited circumstances, to a limited array of public water systems, for a limited period of time.

a. The affordability doctrine meets arsenic

The affordability doctrine was tested in 2001 when the Agency promulgated a revised NPDWR for arsenic (USEPA 2001b). The Agency determined that aggregate estimated annualized benefits of $170 million justified aggregate estimated annualized costs of $210 million, with unquantified benefits making up the difference. USEPA's benefit estimates were contested; for example, Burnett and Hahn (2001a, b) re-estimated them to be only $20 million.

The Agency's cost estimate also was controversial (USEPA 2001a, pp. 7038-7041). For a 10 μg/L MCL, Frey et al. (1998, Table 5) estimated annualized national compliance costs at $708 million, later revised downward to $495 million (Frey et al. 2000, Table 4.8). Gurian et al. (2001) estimated annualized national compliance cost at $294 million, with 90th percent confidence intervals ($177–$495 million) spanning the USEPA and Frey et al. (2000) estimates. Estimates of the national annualized incremental compliance cost for 10 μg/L instead of 20 μg/L include USEPA's $129 million estimate (USEPA 2001a, Table III.E-1), $174 million by Gurian et al. (2001, Table 4), $360 million by Frey et al. (2000, Table 4.8), and $550 million by Frost et al. (2002, Table 1). This disagreement did not subside after promulgation of the final rule (Tiemann 2006; Hilkert Colby et al. 2010; Gingerich et al. 2017).

Congress reacted swiftly to stakeholder concerns about economic feasibility, noting that the revised standard would impose "significant financial costs on many small communities, and many of these communities may find it impossible, because of the financial burden, to be in compliance by 2006 as the rule requires." Members on the appropriations conference committee were "concerned that, because of their complexity, the current waiver and exemption provisions ... may not provide sufficient flexibility for the small communities to receive additional time to reach compliance." They also were "very concerned that ... very small communities may abandon their municipal systems in favor of untreated and unregulated private wells which could create significant other health risks for these communities." They directed USEPA to, by March 1, 2002, "review the Agency's affordability criteria and how small system variance and exemption programs should be implemented for arsenic," taking account of the "undue economic hardship" faced by communities served by small public water systems." But the conferees also hedged their bets, averring that they "do not intend to create loopholes in the Safe Drinking Water Act for compliance to a national arsenic standard" (U.S. House of Representatives 2001, pp. 174-175).

b. USEPA's proposed revision to the affordability doctrine

In its report to Congress (USEPA 2002), USEPA defended the affordability doctrine but did not include the review Congress had directed be conducted. Instead, the Agency briefly summarized public comments received on the 2000 proposed arsenic rule and promised an "ongoing affordability review" that would include soliciting advice from the Science Advisory Board and obtaining "input from stakeholders" (p. 11). Two USEPA advisory committees weighed in with conflicting recommendations (U.S. EPA Science Advisory Board 2002; U.S. EPA National Drinking Water Advisory Council 2003 [including a minority report]).

In 2006, USEPA proposed a revision to the affordability doctrine (USEPA 2006c). The proposed changes would have had mixed or ambiguous effects, most notably the relaxation of the 2.5% MHI regulatory budget. Doctrinally, the proposal reiterated the Agency's longstanding view that economic feasibility is limited to "what may reasonably be afforded by large metropolitan or regional public water systems" (p. 10682), thus leaving unaddressed the economic inefficiency and inequity discussed in Section 3 below. The proposed revision was widely criticized by public commenters (USEPA 2006a) and subsequently by others (Rubin et al. 2007; Crawford-Brown et al. 2009; Raucher et al. 2011; U.S. Conference of Mayors et al. 2013a; U.S. Conference of Mayors 2014). No revised guidance has been finalized.

c. Affordability v. economic feasibility in practice

USEPA has promulgated a number of drinking water rules since SDWA 1996, but in no case has the Agency clearly defined economic feasibility.[2] Some inferences can be made from the Stage 2 Disinfectants and Disinfection Byproducts Rule (Stage 2 DBPR), in which the Agency stated that "it may not be economically feasible for some small systems to install and operate an on-site [granular activated carbon] reactivation facility" (USEPA 2006b, p. 413). USEPA estimated the 50th, 90th, and 95th percentile costs for small systems at $18, $169 and $198 per household served (USEPA 2006b, p. 459, Table VI.E-1). Thus, it might be inferred that USEPA believes the latter two average household-level costs are *not* economically feasible. However, because the Agency did not report benefit estimates by system size, apparent statements about economic feasibility actually may be about affordability.

[2] Two final rules mention the similar term *economically feasible*, but in neither case is the term explicitly defined. See USEPA (1998c, p. 69400 [a restatement of statutory language] and p. 69412 [a declarative statement that a particular standard is "technically and economically feasible"]); and USEPA (2006b, p. 413 [stating that a particular technology "may not be economically feasible for some small systems to install and operate"]).

Inferences also cannot be readily derived from pre-SDWA 1996 rulemakings. For example, in 1994 USEPA re-proposed a NPDWR for sulfates and said that four options "would be economically feasible" despite the financial difficulties presented to small systems (USEPA 1994, p. 65597). With respect to an option costing on average $287 to $811 per household, depending on system size, the Agency acknowledged "concerns that this option would not be economically feasible for small systems" (Table 9). As before, because benefits are absent from the discussion, these statements may be about affordability, not economic feasibility. Further, any inference about USEPA's intentions must be tempered by the knowledge that the re-proposed standard was not promulgated.

Nothing van be inferred from USEPA statements about feasibility without a modifier. The Agency interprets any unmodified use "in terms of technological limits of concentration removal (i.e., treatment) and analytical methods (i.e., sampling and measurement)" (Raucher and Cromwell 2004, p. 2) – i.e., without economic content. In short, it appears that a general principle of *economic* feasibility is ground that USEPA has never plowed in the context of drinking water because it was supplanted by the affordability doctrine.

But the affordability doctrine appears to have reached its practical limit, as the cost of drinking water now approaches (and may even exceed) the 2.5% MHI regulatory budget. Raucher and Cromwell (2004) relied on USEPA RIAs to build their best estimates of aggregate compliance costs of SDWA 1996 regulations ($1.8 billion per year) and seven regulations promulgated under SDWA 1986 ($4.8 billion per year) (both $2004; 7%). Household-level equivalents are generally not available because they are absent from the RIAs. Nonetheless, household-level estimates may breach the affordability threshold. In the case of the Arsenic NPDWR, for example, Raucher and Cromwell (2004) conclude that "average households in the

smallest system size category will pay at least 14 times as much (on average) as households served by utilities in the largest size categories," with annual compliance cost per household exceeding $700 (pp. 20-21).

Other analyses have obtained similar results. In a recent California study, the median (SD) total cost of drinking water for surveyed communities was estimated at $1,172 ($488) per household, with costs in many jurisdictions exceeding the 2.5% MHI regulatory budget (U.S. Conference of Mayors 2014, pp. 4, 12-13, Table C). At some point, it may not be possible for USEPA to promulgate new NDPWSs without unambiguously violating its affordability doctrine.

3. Inefficiency, Inequity, and Arbitrariness in Federal Drinking Water Regulation

It is therefore not surprising that many NPDWRs are economically inefficient. From the outset, SDWA directed USEPA to set stringent national standards while providing ways for small systems to escape the higher cost of complying with standards that were not economically feasible. But the Agency emphasized regulatory stringency and provided limited accommodation to small systems (Schnare 1998). This led Congress to require benefit-cost balancing in SDWA 1996.

SDWA regulations also are inequitable. They impose significantly disproportionate costs on non-metropolitan areas, low-income communities, and low-income households generally. The tradeoff between efficiency and equity, made famous by Okun (1975), does not seem to apply.

a. Inefficiency

NPDWRs are economically inefficient at least four ways. First, as noted earlier, USEPA sets standards based on what it determines to be economically feasible for very large public water systems (USEPA 2006c, p. 10682). Because economies of scale in drinking water supply

and treatment are so strong, standards that are economically feasible for very large systems are virtually certain to be economically infeasible for small systems.

Second, neither variability across systems within a system-size category nor household variability within each system is considered (USEPA 2006c, p. 10673). Thus, even if a NPDWR were economically feasible for the median system in a system-size category, it would be economically infeasible for as many as half of the systems in that category, and as many as half of the households in the median system.

Third, USEPA estimates compliance expenditures, not opportunity costs. At the household level, this means not counting the benefits foregone from reduced expenditures on food, shelter, education, health care, transportation, and expenditures on goods and services that improve health (Keeney 1990, 1994; Lutter and Morrall III 1994; Lutter et al. 1999; Cory and Taylor 2017). At the system level, expenditures to comply with economically infeasible NPDWRs reduce resources available for infrastructure investments, the need for which has been estimated to exceed $1 trillion over 25 years (AWWA 2012). This includes extensive replacement of drinking water service lines to reduce or eliminate lead exposure, ironically to comply with a different and notoriously expensive NPDWR. When net social benefits are estimated, they are upwardly biased because expenditures, not opportunity costs, are subtracted from estimated benefits.

Fourth, the affordability doctrine does not take account of benefits, and even if benefits were counted, USEPA's estimates tend to be upwardly biased by the Agency's longstanding practice of "policy framing" risk estimates at the "upper end" (U.S. EPA Office of the Science Advisor 2004). Thus, even if opportunity costs were properly estimated and subtracted from benefits, estimates of net social benefits would be upwardly biased.

Evidence of economic inefficiency extends back to early SDWA 1974 regulations, when several economic inefficiencies were identified (Council on Wage and Price Stability 1978) in USEPA's 1978 proposed trihalomethane NPDWR (USEPA 1977). The standard was estimated to cost on average $3.9–6.3 million ($1978) per excess cancer case prevented, depending on stringency, but exceed $12 million per case for water systems serving 75,000 or fewer persons. These inferences were confirmed and extended to other NPDWRs in subsequent analyses (Morrall III 1986; Office of Management and Budget 1991; Raucher et al. 1994).

b. Inequity

Households served by small systems are assured of being treated inequitably multiple ways. First, the affordability doctrine imposes disproportionate burdens on lower-income households (U.S. EPA Science Advisory Board 2002, Berahzer 2012; Teodoro 2018). Households with income below the MHI must pay more than 2.5% of their income for drinking water.

Second, USEPA ignores regional and State differences in MHI. Median households in regions and States with lower MHIs therefore must pay more than 2.5% of their income for drinking water. National MHI in 2016 was $58,820 (90% CIs: ± $102), but ranged from $40,528 (90% CIs: ± $258) in Mississippi to $72,935 (90% CIs: ± $1,164) in the District of Columbia – a range of 1.8x. About 60% of Mississippi households earned $50,000 or less compared to 38% of District households (U.S. Census Bureau 2016).

Third, communities with below-average incomes must bear costs that the Agency regards as unaffordable by households served by large metropolitan systems. MHI in all 82 Mississippi counties was below the national MHI in 2016. In 20% of Mississippi counties, MHI was less than half of the national MHI (U.S. Census Bureau 2016). The affordability doctrine requires

that they commit 5% of income, or more, to drinking water. This is equivalent to a peculiar regressive tax that, instead of producing government revenue, transfers wealth from the poor to firms in the water treatment industry.

c. Arbitrariness

The affordability doctrine relies on other arbitrary elements besides the 2.5% national MHI threshold, and thus results in outcomes that are inherently arbitrary on other margins as well (U.S. EPA Science Advisory Board 2002; Irvin 2017). Each of the 151,000 water systems in the U.S. (USEPA 2017) could be defined as its own domain or aggregated into a single domain for standard-setting. In practice, water systems are aggregated into arbitrarily defined system-size categories in the expectation that variability is captured by economies of scale. But as Gingerich et al. (2017) showed for small systems participating in an Agency-sponsored demonstration project exploring arsenic treatment alternatives, direct costs can be highly variable for reasons other than system size. The median annual treatment cost among systems in this demonstration project was $147 per household, but ranged from $17 to $662 per household. The median increase in total water bill was 55%, but ranged from 6% to 274%.[3] These ranges suggest that the arsenic MCL may not be technologically feasible for small systems "under field conditions," as required by SDWA 1996 § 1412(b)(4)(D).

Similarly, there is nothing special about MHI as an affordability metric, nor does the particular MHI statistic obtained from the Census Bureau lack critics (see, e.g., Eskaf 2013; Irvin 2017). Several alternative approaches have been suggested to make the affordability doctrine sensitive to the disproportionately high opportunity costs experienced by low-income households

[3] Figures derived from Gingerich et al. (2017), Table 3.

(Raucher et al. 2011; U.S. Conference of Mayors et al. 2013a, b; UNC Environmental Finance Center 2017; Teodoro 2018, 2019; Czerwinski et al. 2018; Raucher et al. 2019). USEPA did not propose to adopt any of the earlier recommendations but may be considering the more recent ones.

d. USEPA's policy of ensuring equal *ex post* health risk from drinking water imposes highly unequal *ex post* risk elsewhere

Equal protection can be defined as ensuring that people receive the same quantity at different prices or pay the same price and receive different quantities. Quantity-based equal protection is reasonable and appropriate for the provision of constitutional rights (e.g., free speech, protection from unreasonable search and seizure, trial by jury) and public goods supplied by government and funded by general taxation (e.g., national security). However, drinking water is a private good even where it is provided by a public entity. Consumption is rivalrous, and sellers can deny access to those who refuse to pay.

Drinking water is more like other private goods and services, such as electricity, natural gas and sewage treatment, that often are supplied by regulated public or private monopolies because of high fixed costs. Where risk reduction is an attribute of a private good or service, consumers typically pay the same price but obtain different quantities of risk reduction depending on other factors, such as their propensity for risk-taking. The result is variability in *ex post* risk outcomes, not *ex post* price differences. Indeed, in some jurisdictions, public utilities are statutorily required to charge all customers the same rate (Berahzer 2012; U.S. EPA Environmental Financial Advisory Board 2016; UNC Environmental Finance Center 2017), a practice that is consistent with *price*-based equal protection.

A few examples from outside the world of public utilities provide additional insight. For decades, automobile manufacturers marketed as options various safety features, such as seat belts, air bags, and antilock brakes, charging the same price to all similarly situated customers. But customers varied greatly in the amount of risk reduction they obtained, largely due to fixed differences in baseline risk and their propensity to engage in risky behavior. The same pattern applies to newly invented automotive safety features, such as lane-departure, forward-collision, and blind-spot warning technologies. The unit price of these technologies is fixed, at least in broad categories, but the quantity of risk reduction obtained varies across consumers. Persons who are relatively risk-averse, or have reason to believe that they are riskier than average, are more likely to purchase these safety innovations. Some consumers also are more likely to adapt their behavior in ways that reduce risk-reduction benefits (Peltzman 1975). Regardless of their baseline risk, preference for risk-taking, or propensity to engage in adaptive responses that reduce benefits, they still pay the same price.

There is a vibrant consumer market for inherently risky power tools. Operating risk varies greatly because consumers differ in baseline conditions (e.g., experience, technical skill, intensity of use) and their propensity to engage in risky behavior (e.g., read and follow directions, wear protective clothing and equipment). Still, all consumers pay the same price for each risk-reduction technology built into these products. What varies across consumers is the quantity of risk reduction obtained.

This pattern applies even to speculative benefits. The establishment of the National Organic Program by the Agricultural Marketing Service of the U.S. Department of Agriculture gave food producers a government-sanctioned way to appeal to consumers who believe that foods certified as organic are safer than conventional foods. Consumers who purchase certified

organic foods pay the same, higher unit prices regardless of how much psychological benefit they receive.

It is difficult to find examples of private goods with risk-reducing attributes that deliver the same *ex post* risk but, like drinking water under EPA's quantity-based equal protection policy, cost consumers widely different amounts. The typical consumer experience is one of variable *ex post* risk resulting from differences in baseline risk, risk preferences, and the net quantity of risk reduced – not differences in the price of the risk-reducing attribute or feature.

Since 1974, USEPA has implemented SDWA to ensure the same level of *ex post* protection from drinking water contaminants (Schnare 1998), not the same price for risk reduction. The practice appears to have been first codified as policy during the National Performance Review (NPR), a management effort to streamline Federal government operations to improve performance and reduce costs (Gore Jr. 1993, 1994). Although official NPR reports are silent on equal protection, USEPA appears to have leveraged the NPR to make quantity-based equal protection as Agency policy (USEPA 1996).[4]

By insisting on quantity-based equal protection for potential health risks, USEPA imposes highly unequal protection from actual financial risks that inevitably result from highly unequal prices (Raucher 2003).[5] Households served by small systems cannot obtain the same

[4] See p. 2: "The basic concept of environmental protection has evolved beyond just pollution control to include broader issues, such as pollution prevention, sustainability, and environmental justice--e.g., businesses are looking to cut waste in order to prevent pollution and improve profitability; government agencies are developing incentives that can lead businesses and communities to go beyond compliance and continuously improve environmental performance; *citizens demand government policies that ensure equal protection against environmental risks* and create economic opportunity for present and future generations" (emphasis added).

[5] See p. 232: "While a uniform MCL for small and large systems does provide households with roughly 'equal health protection' in terms of exposure, regardless of where they live or the size of the community water system (CWS) that serves them, the cost burden that each system bears can be significantly different. In essence, any given MCL is likely to impose much higher costs per unit of risk reduction benefit received by households served in small systems relative to the costs per risk reduction borne in larger communities. This raises a fundamental issue of fairness — should families served by small systems be forced through regulations to pay much higher costs for their risk reduction benefits than do households in larger, more urban settings?"

quantity of *ex post* health risk except by bearing disproportionately large financial risks. Similarly troubling is the highly unequal protection from financial risk imposed on non-metropolitan areas, low-income communities, and low-income households generally, that inevitably results from compulsory quantity-based equal protection from potential health risks. Environmental justice advocates have challenged differential health protection in drinking water as a violation of Title VI of the Civil Rights Act, but these efforts have not succeeded for lack of discriminatory intent (Lado 2017). But the case for intentional discrimination is stronger with respect to regulatory policies that, like USEPA's practice in SDWA, knowingly impose disproportionate financial risks.

4. A Rational Economic Interpretation of the SDWA Statutory Framework

SDWA as amended can be interpreted as providing the rational economic framework for standard-setting that was missing from SDWA 1974 and SDWA 1986. For any given contaminant or constituent, SDWA 1996 directs USEPA to promulgate NPDWRs that are "as close to the maximum contaminant level goal as is [technologically] feasible" (SDWA 1996 § 1412(b)(4)(B)). But at the same time, the Agency must determine "whether the benefits of the maximum contaminant level justify, or do not justify, the costs" (SDWA 1996 § 1412(b)(4)(C)) after an objective analysis of risks, benefits, and costs (SDWA 1996 § 1412(b)(3)). USEPA could interpret these texts as requiring the promulgation of NPDWRs that minimize health risk subject to the twin constraints of technological and economic feasibility, and account for system size, source water, and other key factors in determining both technological and economic feasibility.

a. Pre-SDWA 1996: Deadweight Losses and Inequity through Selective Economic Feasibility

For all but the largest metropolitan water systems, NDPWRs have been economically infeasible by design (USEPA 2006c, p. 10673). Variances and exemptions were theoretically available for small water systems (Schnare 1998), but relief if granted was temporary, and systems were generally required to comply with standards that were less stringent than NPDWRs but still economically infeasible. That is, systems for which NPDWRs were economically infeasible have no option except to comply by making their customers poorer.

This is illustrated in Figure A. Assume six alternative MCLs ranging from MCL_a (the most stringent) to MCL_f (the least stringent), and three different sizes of public water systems (Large [L], Small [S] and Very Small [VS]), which for simplicity in exposition are all presumed to have the same baseline level of a contaminant. Economies of scale in water treatment result in three different marginal cost schedules (MC_L, MC_S, and MC_{VS}) and the common marginal benefit schedule MB. All six alternative MCLs are technologically feasible for all three system sizes, so USEPA would have chosen MCL_b as the NPDWR. MCL_b is economically feasible for Large systems because the marginal benefit of treatment equals the marginal cost ($MB = MC_L$). Households served by Large systems would pay a price of P_L.

Though it is technologically feasible for Small systems to achieve MCL_b, they can only do so at the higher household price P_S. Households served by Small systems therefore are unambiguously worse off. At the margin, they pay P_S to obtain benefits worth only P_L. The difference in expenditure, $P_S - P_L$, multiplied by the quantity of drinking water consumed, is transferred from their family budgets to firms that design, manufacture, install, or operate water treatment technology. They suffer a deadweight loss equal to area A.

The situation is worse for households served by Very Small systems. They must pay P_{VS} for benefits worth only P_L. They must transfer even more income to firms in the water treatment industry. They suffer the deadweight loss of areas *A* plus *B*.

These inefficiencies have been defended on equity grounds by defining equity in terms of a quantity-based definition of equal protection. While it is true that all households gain the same *ex post* quantity of health risk, households served by Small and Very Small public water systems obtain this only by being made permanently poorer. USEPA achieves quantity-based equal protection from health risk by reducing the welfare of those it purports to benefit.

[[INSERT FIGURES A AND B ABOUT HERE]]

b. Achieving efficiency and reducing inequity through economic feasibility

Interpreting SDWA 1996 as a problem of risk minimization subject to technological and economic constraints allows economic inefficiency and price-based unequal protection to be substantially reduced. This can be accomplished by setting NPDWRs that are both technologically *and* economically feasible for the smallest size systems subject to regulation. That is, USEPA would interpret SDWA 1996 §§ 1412(b)(4)(B)-(C) as applying consistently to all water systems subject to Federal standards, not just the handful of metropolitan systems large enough to spread compliance costs over hundreds of thousands or even millions of households. The structure of the amended statute arguably is more consistent with this approach, which makes national standards a flexible floor instead of a *de facto* ceiling. States can choose to set more stringent standards, but they would have to transparently acknowledge, and take

responsibility for, the resulting deadweight losses, reverse Robin Hood income transfers to the drinking water treatment industry, and the inequities of quantity-based equal protection.

Figure B shows how this would work. Very Small systems would be exempt. The MCL would be set at MCL_d, the most stringent standard that is economically feasible for Small systems. Households served by Large and Small systems would pay prices P_L* and P_S*, respectively. The deadweight loss in Figure A would be avoided.

Initially, Large and Very Small systems would suffer the deadweight losses in areas *C* and *D*, respectively, because in both cases there would be uncaptured net benefits from more stringent controls. But these deadweight losses likely would be temporary. Households served by Large systems would be willing to purchase treatment technology sufficient to achieve MCL_b, the most stringent standard that is economically feasible for them. These households would pay $P_L^{"}$ instead of the lower price P_L^*, but the marginal value of additional health benefits would exceed the price difference. Similarly, even though Very Small systems would be exempt, their customers would be willing to purchase treatment technology sufficient to achieve MCL_f, the most stringent standard that is economically feasible for them. Households served by Very Small systems would be willing to pay $P_{VS}^{"}$ to gain the value of additional health benefits, which exceed marginal cost for every unit of contaminant greater than MCL_f.[6]

These choices are rational because both groups of households would be better off than under MCL_d. No regulatory coercion is needed to motivate Large or Very Small systems to voluntarily achieve standards more stringent than MCL_d. Firms in the drinking water treatment

[6] In theory, a water system could respond strategically by breaking up into smaller units in order for each part to belong to a smaller system size category. If the purportedly smaller systems shared management control and, more importantly, treatment technology, States likely would recognize such responses as shams and deny them regulatory approval. On the other hand, both efficiency gains and inequity reductions might be realized in cases where water systems (typically those relying on groundwater) are inefficiently large as a result of prior efforts to comply with NPDWRs that disadvantage small systems. In these cases, subdivision might well be justified.

industry would gain new incentives to invest in research and development that reduces the cost of serving the needs of Very Small systems instead of lobbying for Federal standards that are profitable for them but financially damaging for Very Small system customers.

This solves the allocative efficiency problem because NPDWRs would not impose an economically infeasible drinking water treatment technology on anyone. Price-based unequal protection would remain, however, because households would pay prices inversely related to the size of the system serving them. But these price differences would reflect differences in cost of service, which generally are not perceived as a discriminatory practice in drinking water supply (UNC Environmental Finance Center 2017, pp. 17-18).

The only other alternative that solves the allocative efficiency problem and achieves similarly equitable results requires EPA to set multiple MCLs within a single NPDWR.[7] This might be compatible with SDWA 1996, but it would have other significant drawbacks. First, it would require the Agency to obtain much more extensive knowledge across tens of thousands of water systems in the United States. This would be expensive and undoubtedly require a significant expansion in the ranks of its technical staff. Second, the task would be duplicative. The only logical sources for this information are local water systems. It is suboptimal to recycle local information through USEPA and return it as a regulation imposed on the same systems from which the data were obtained. A suboptimal Federal information recycling loop might make sense if systems lacked sufficient expertise or faced perverse incentives to analyze their situations incorrectly. But systems have access to a wealth of technical expertise elsewhere (e.g., the American Water Works Association and the National Rural Water Association, among

[7] The option of setting dual standards is discussed in U.S. EPA National Drinking Water Advisory Council (2003), and U.S. EPA National Environmental Justice Advisory Council (2009).

others), and the incentives they face are compatible with allocative efficiency, minimizing price-based unequal treatment, and responding to customer concerns.

5. Effects of Adopting the Economic Feasibility Principle

Predictions about the effects of the economic feasibility principle on future USEPA actions are inherently speculative. There is limited public information concerning which contaminants the Agency expects to advance to rulemaking. The Agency has disclosed, via the Fall 2018 Semiannual Regulatory Agenda, that it intended to propose revisions to the Lead and Copper NPDWR (RIN 2040-AF15) and a new NPDWR for perchlorate (RIN 2040-AF28) (Regulatory Information Service Center 2018), and it has sought comment on three proposed regulatory options for perchlorate (USEPA 2019). However, insufficient information has been publicly disclosed to allow even a preliminary determination of which alternatives, if any, may be both technologically and economically feasible.

The economic feasibility principle may receive its first test elsewhere. The California State Water Resources Control Board (SWRCB) is developing guidelines or regulations defining economic feasibility. This effort follows the remand and vacatur of a drinking water standard for hexavalent chromium (*CMTA et al. v. SWRCB* 2017). The court interpreted the California SDWA, which is modeled on the Federal SDWA, in a manner that is theoretically very similar to the economic feasibility principle proposed here, except that it left to the SWRCB the task of defining the term.[8]

[8] "[T]he court interprets the [California] Safe Drinking Water Act as requiring the Department to set the MCL at a level that is *as close as economically feasible to*" the Public Health Goal. *CMTA et al. v. SWRCB*, p. 6 (emphasis added).

Would the adoption of the economic feasibility principle after SDWA 1996 have resulted in reduced inefficiency and inequity? Some insight can be gleaned by reviewing post-SDWA 1996 regulations and attempting to discern whether USEPA would have chosen different regulatory alternatives. Retrospective inferences of this sort are necessarily limited to the economic analyses the Agency performed, and different alternatives likely would have been examined if economic feasibility had been an explicit regulatory objective. For this reason, any retrospective look at past regulations should be viewed as suggestive rather than dispositive.

Since 1974, USEPA has promulgated NPDWRs for seven microorganisms, three disinfectants, four disinfection byproducts, four radionuclides, 16 inorganic chemicals, and 53 organic chemicals (USEPA 2018b). This section summarizes the extent to which regulations promulgated since 1996 were economically feasible, and if not, whether an economically feasible alternative was considered. Further information is provided in the Online Appendix.

Table 1 lists the most relevant post-SDWA 1996 regulations, and reports whether (a) USEPA determined that benefits justified costs and (b) the economic analysis identifies an option that would have been preferred under the economic feasibility principle. Only a few regulations meet both criteria; some do so, but inferences about alternative regulatory choices are still elusive. Details are summarized here; more information about the economic feasibility of these regulations is provided in the Online Appendix.

[[INSERT TABLE 1 ABOUT HERE]]

a. Surface water treatment regulations

A suite of surface water treatment rules targeting *Cryptosporidium* (USEPA 1998d, 2001a, 2002b, 2006b) depended on benefit estimates obtained by risk modeling. These rules were determined to be economically feasible, but modeled illnesses and deaths averted substantially exceeded the number of illnesses and deaths reported from all pathways by the Centers for Disease Control and Prevention (CDC).[9] Because the model was not validated and the substantial gap between modeled and reported illnesses and (especially) deaths was not adequately explained, it is reasonable to be skeptical about economic feasibility claims. It is possible that some or all of these rules were economically feasible for very large systems, where economies of scale yield low marginal costs. However, the gap between modeled and reported illnesses and (especially) deaths is so great that even this inference is doubtful. Moreover, it is highly unlikely that any of these rules were economically feasible for smaller systems.

b. Disinfection byproduct regulations

Two regulations were promulgated to manage risks from disinfection byproducts (DBP) (USEPA 1998c, 2005). Any determination of economic feasibility depends on the assumption that mixed, weak, and unconfirmed epidemiological associations between DBP and cancer are causal. USEPA acknowledged that estimated cancer risks could be zero but nevertheless assumed causality. Without this assumption, neither regulation would have been economically feasible, even for very large systems.

[9] See Online Appendix, Section A1(h).

c. Specific contaminant NPDWRs

For radionuclides, USEPA retained then-existing MCLs for combined Ra^{226} and Ra^{228} (though with intensified monitoring requirements), beta particle and photon emitters, and gross alpha activity. It declined to set a more stringent MCL.

For uranium, USEPA's estimates of aggregate net benefits were negative at every analyzed alternative. Thus, even the least stringent alternative analyzed (80 μg/L) was too stringent to be an economically feasible NPDWR. According to USEPA's economic analysis, the promulgated 30 μg/L uranium MCL was estimated to yield about $50 million in negative net benefits, with a marginal cost of $68 million per cancer case averted. Promulgating 80 μg/L instead of 30 μg/L as the MCL would have saved $36.4 million while reducing the number of cancer cases averted by 0.35, a marginal cost of $190 million per cancer case. It is possible that 80 μg/L (or 30 μg/L, the MCL promulgated) would have been economically feasible for very large systems. However, uranium is not a significant constituent in source waters used by very large systems, so it is probable that the rule was economically feasible for no water system.

Similarly, the arsenic MCL (10 μg/L) was estimated to yield negative aggregate net benefits at every MCL examined. The MCL was economically feasible for systems serving ≥ 1 million persons under fairly broad conditions, however. Under several restrictive conditions (3% discount rate, upper-bound arsenic exposure, average household size ≥ 5), the MCL may have been economically feasible for the 10,001–50,000 person system-size category. These restrictions are so significant – especially the minimum household size – that no systems in this size category probably qualifies.

The 1991 lead and copper rule (LCR) was estimated by USEPA to cost systems $460–$750 million per year and States $40 million per year (USEPA 1991, Table 20). USEPA did not

report benefit and cost estimates for the 2000 LCR revision (USEPA 2000a), so no inferences can be gleaned concerning the specific effect of the economic feasibility principle had it been adopted.[10] Abernethy et al. (2018) addresses the high cost of simply determining where lead service lines exist and developing a cost-effective strategy for line replacement. and the USEPA Science Advisory Board (2011) has raised concerns that service line replacement may increase rather than reduce risk. This has obvious relevance under the economic feasibility principle, but it is immaterial if technological feasibility alone controls decision-making. USEPA's forthcoming Long-Term LCR (USEPA 2018a), which purports to overcome limitations that made the 2000 regulation ineffective, has been estimated to cost another $44–274 million per year (Slabaugh et al. 2015, Table 6), with annual costs rising exponentially as system size declines (Slabaugh et al. 2015, Figure 5).

d. Summary

If USEPA had adopted the economic feasibility principle after SDWA 1996, the entire portfolio of SDWA 1996 regulations would have looked very different. USEPA would have set MCLs such that they were economically feasible for the smallest system size subject to regulation, making tradeoffs between stringency and regulatory scope. These tradeoffs would have been more striking if the Agency also had complied with the relevant SDWA 1996 provisions governing risk assessment and benefit-cost analysis.

Allocative inefficiency would have been reduced or avoided to an extent dependent on the objectivity of USEPA's risk and economic analysis. Cost-savings would have totaled hundreds of millions of dollars annually. These savings would have enabled water system

[10] USEPA acknowledged in the 1991 LCR that it did not take benefits into account in setting the NPDWR. See USEPA (1991).

managers to better fund key infrastructure improvements, and households would have had greater disposable income for more highly valued uses.

Post-1996 drinking water regulations, like those which preceded SDWA 1996, imposed gross inequities on nonmetropolitan areas, low-income communities, and low-income households. The economic feasibility principle would have been substantially attenuated these inequities. The poor would not have borne disproportionately high and rising costs for drinking water, and drinking water regulation would not have made them poorer with each successive regulation action.

6. Conclusion

The SDWA has been implemented in ways that cause significant allocative inefficiency by subordinating economic feasibility to stringent standards regardless of health benefits. Regulations also have achieved quantity-based equal protection against health risks at the cost of substantial unequal protection against economic and financial risks, most notably on households served by small systems, less wealthy regions and States, low-income communities, and low-income households everywhere.

In SDWA 1996, Congress changed direction and mandated that USEPA promulgate standards supported by objective risk assessment and benefit-cost analysis and costs are justified by benefits. Ceteris paribus, adoption of the economic feasibility principle would have led to less stringent MCLs, the exemption of smaller system sizes, or both. How USEPA would have traded off stringency for regulatory scope cannot be determined because policy choices delegated to the Administrator can be informed but not determined by economic analysis.

It also appears that post-SDWA 1996 NPDWRs did not adhere to SDWA 1996 requirements for objective risk assessment or Agency guidance on economic analysis (USEPA

2000b, 2008, 2010). Each determination that benefits justified costs depended on estimating expenditures instead of opportunity costs, using risk assessment methods that were not designed to be compatible with benefit-cost analysis, or invoking speculative co-benefits to make up any difference between estimated benefits and estimated expenditures.

USEPA can adopt the economic feasibility principle by regulation as the most reasonable interpretation of SDWA 1996. The Agency can set MCLs at the most health-protective level consistent with technological and economic feasibility for the smallest size system subject to its regulation. Systems that can achieve more stringent controls in an economically feasible manner will be motivated to do so without a Federal mandate. This includes both large water systems (for which even very stringent standards may be economically feasible because of economies of scale in treatment) and very small systems (which would be exempt because of their diseconomies of scale). The economic feasibility principle would balance customers' interests in protection from risk and their aversion to expenditures that exceed the value of benefits received.

In the economic analysis of a USEPA regulation outside the drinking water realm, Wagner (2009) detected a material bias favoring the Agency's preferred standard rather than a fair comparison of alternative that could have informed the Administrator's decision. Many of the economic analyses for post-SDWA 1996 regulations are consistent with Wagner's observations, and none of the analyses contradicts it. This means adoption of the economic feasibility principle would be an essential prerequisite for economically feasible standard-setting, with the courts serving as arbiters of compliance with SDWA 1996 procedural requirements. Still, even if adopting the economic feasibility principle is not sufficient to achieve it, there should be no doubt that its adoption is essential.

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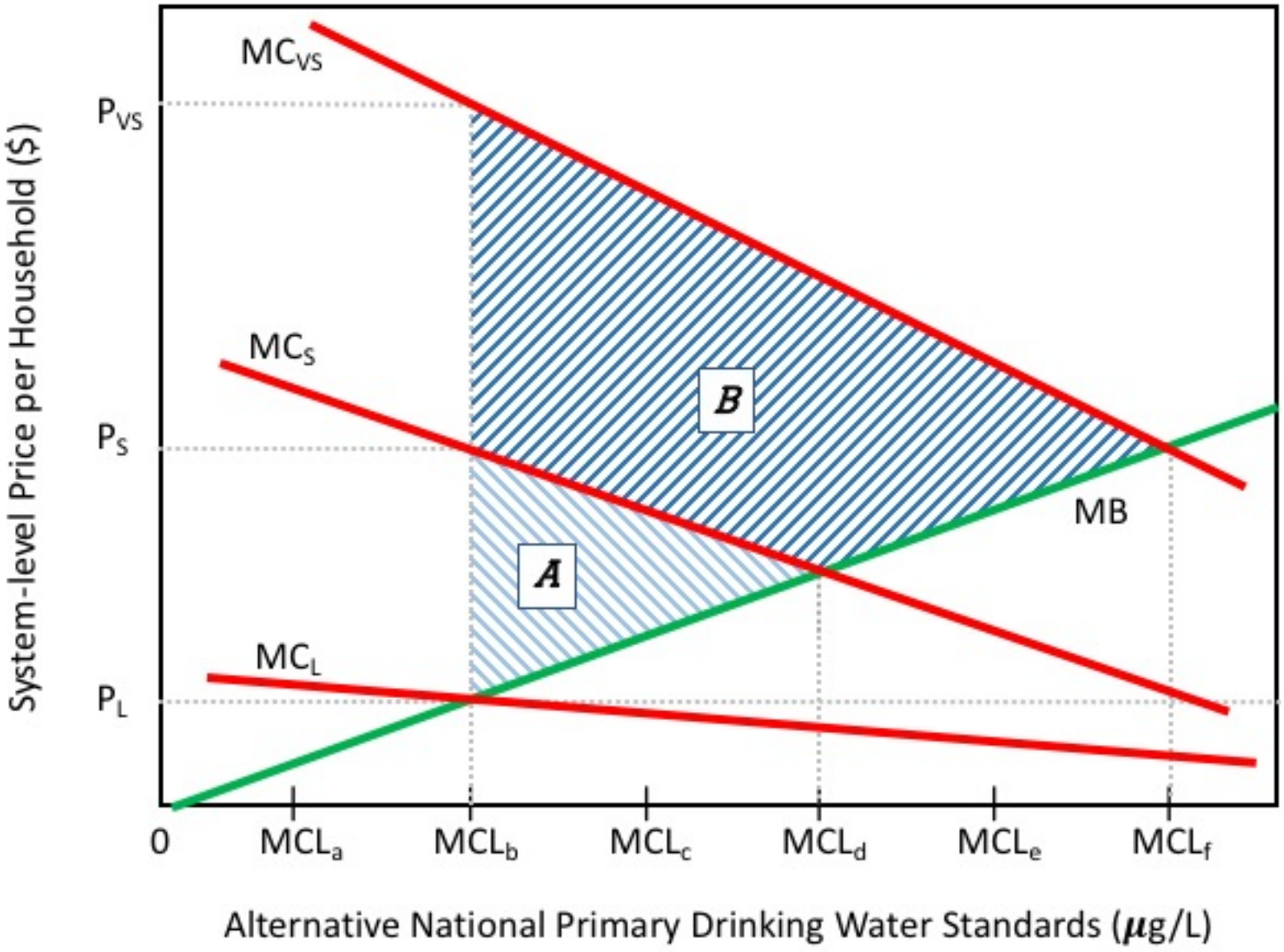
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System-level Price per Household ($)
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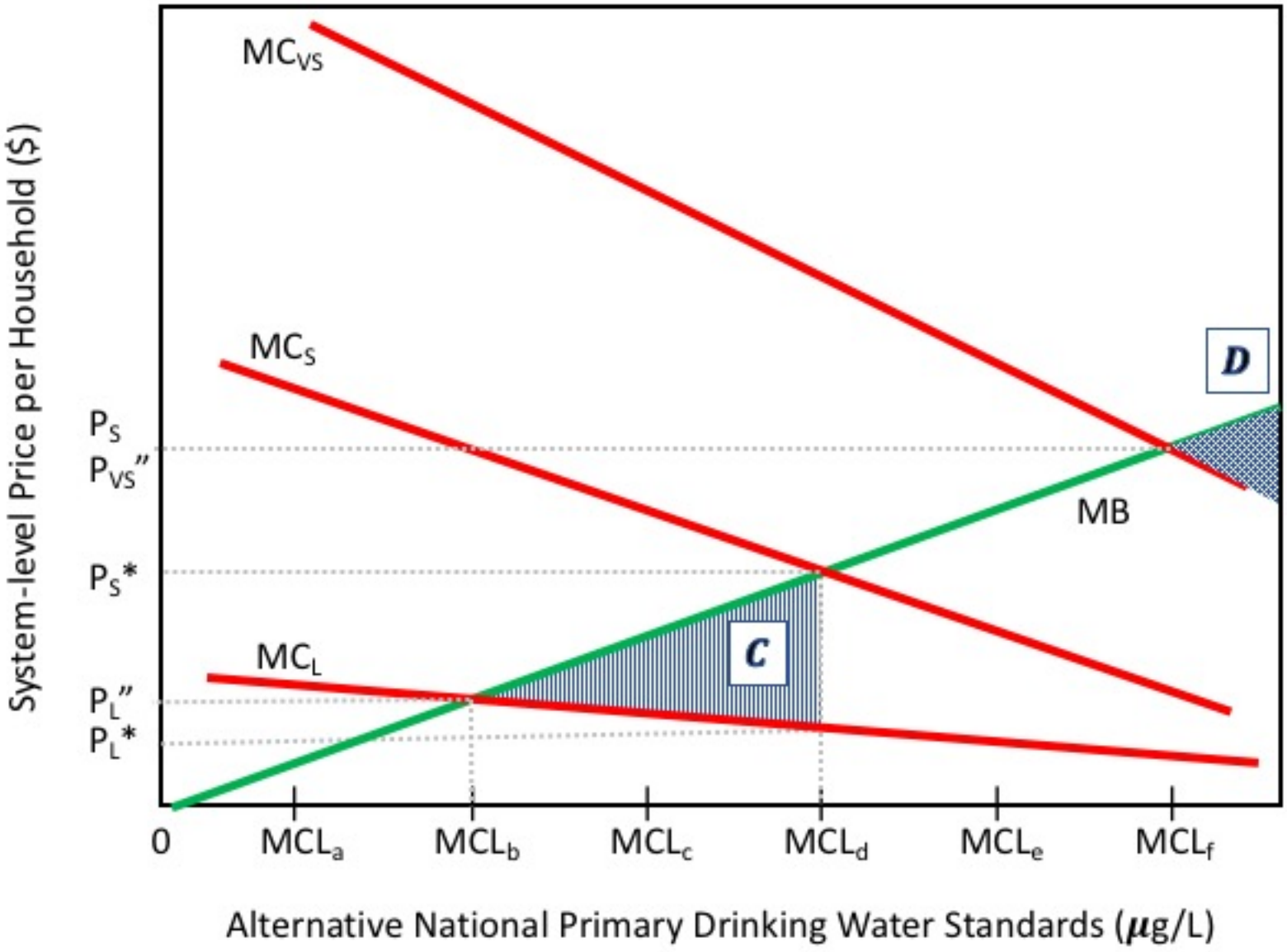
Alternative National Primary Drinking Water Standards (μg/L)


System-level Price per Household ($)
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Alternative National Primary Drinking Water Standards (μg/L)

Table 1: Post-SDWA 1996 Rulemakings and Economic Feasibility

Post-SDWA 1996 Rulemaking	Date/ Reference	"Benefits Justify Costs"	Economic Feasibility Can Be Reasonably Characterized Based on USEPA Analysis
Disinfectants and Disinfection Byproducts Rule (DBPR)	12/16/1998 63 FR 69390	Yes	No
Interim Enhanced Surface Water Treatment Rule (IESWTR)	12/16/1988 63 FR 69478	Yes	No
Lead and Copper NPDWR	1/12/2000 65 FR 2000	No	No
Revisions to IESWTR, Stage 1 DBPR	4/4/2000 65 FR 20304	No	No
Public Notification Rule	5/4/2000 65 FR 40520	No	No
Radionuclides NPDWR	12/7/2000 65 FR 76708	Yes	Yes
Uranium NPDWR		Yes	Yes
Revisions to IESWTR, Stage 1 DBPR	1/16/2001 66 FR 3770	No	No
Arsenic	1/22/2001 66 FR 6976	Yes	Yes
Filter Backwash Recycling Rule (FBRR)	6/8/2001 66 FR 31085	Yes [a]	No
Long Term 1 ESWTR	1/14/2002 67 FR 1811	Yes	No [b]
Stage 2 DBPR	1/4/2006 71 FR 388	Yes	Yes
Long Term 2 ESWTR	11/8/2006 71 FR 653	Yes	Yes
Ground Water Rule (GWR)	11/8/2006 71 FR 65573	Yes	Yes
Lead and Copper: Short-Term Regulatory Revisions and Clarifications	10/10/2007 72 FR 57782	No	No
Drinking Water Regulations for Aircraft Public Water Systems	10/19/2009 74 FR 53589	No	No
Revisions to the Total Coliform Rule (RTCR)	2/13/2013 78 FR 10270	Yes	No

Notes: [a] "The Agency has determined that the benefits of the FBRR justify their cost on a qualitative basis." [b] Key analytic documents are not publicly available.

Achieving Economically Feasible Drinking Water Regulation

APPENDIX

Richard B. Belzer

This Appendix provides reviews of the economic feasibility of individual post-SDWA 1996 National Primary Drinking Water Regulations (NPDWRs), summarized in Section 5 of the Main Text. Section A1 covers the cluster of regulations dealing with microbial risks, disinfection, disinfection byproducts, and disinfection-resistant pathogens. Section A2 summarizes NPDWRs for radionuclides, arsenic, and lead and copper. Section A3 summarizes miscellaneous drinking water regulations related to routine bacterial pathogens. Only those regulations in which USEPA asserted that benefits justified costs and conducted an economic analysis are included.

This review generally takes as given USEPA's benefit and cost estimates unless alternative estimates were readily available at the time, even when there are good reasons to doubt their accuracy (e.g., due to structural bias) or reliability (e.g., due to excess precision). Certain methodological errors were endemic, such as estimating expenditures instead of opportunity costs and calculating net benefits as the difference between opportunity-cost based benefits and expenditures. These errors are noted but generally ignored here because the objective is to ascertain whether prior adoption of the economic feasibility principle would have resulted in a different decision when all other factors are held constant. Thus, it is assumed that if the Agency had adopted and followed the economic feasibility principle it would not have used different data or analytic methods to estimate benefits and costs. It is further assumed that the relevant economic analyses were not reverse-engineered to "serve primarily as a mechanism for promoting agency decisions rather than scrutinizing them" (Wagner 2009, p. 56).

There is no practical way to review the Agency's work and draw inferences concerning what decisions it might have made under the economic feasibility principle where these assumptions do not hold or if superior analytic methods had been used. Nonetheless, it is clear that if USEPA had adopted and applied the economic feasibility principle, Maximum Contaminant Levels (MCLs) would have been less stringent, the minimum system size subject to regulation would have been larger, or some combination thereof. These effects would have been more pronounced if the Agency had estimated opportunity costs instead of expenditures, and objectively estimated risks and benefits as required by SDWA 1996 § 1412(b)(3)(A).

A1. Microbial/Disinfection Byproducts Rule Cluster

USEPA formed the Microbial-Disinfection/Disinfection Byproducts Advisory Committee (M-DBP) in February 1997 as a follow-on regulatory negotiation (colloquially referred to as a "Reg Neg") to a similar proceeding conducted in 1992-93. The domain of the more recent Reg Neg included certain proposed rules specifically referenced in SDWA 1996 § 1412(b)(2)(C).

The suite of rules in this cluster seeks to address microbial risks through disinfection, countervailing risks posed by disinfection byproducts (DPBs) after disinfection, and microbial contaminants that are resistant to control by disinfection (principally *Cryptosporidium* and *Giardia*). Risks were estimated and characterized in accordance with standard Agency methods designed to ensure that they were not "*unrealistically* conservative" (Browner 1995, p. 3, emphasis added). This is an ambiguous but clearly different approach than required by SDWA 1996 § 1412(b)(3)(A)(i), which states that risk assessments must be objective. For example, in the Stage 1 Disinfection Byproduct Rule (Stage 1 DBPR), USEPA described the scientific evidence that DBPs cause bladder cancer and reproductive and developmental effects as "inconclusive" (USEPA 1998d, p. 5-15). Nevertheless, the Agency assumed that there was an

80% probability that the weak associations with bladder cancer observed in some epidemiological studies were causal (USEPA 1998d, pp. 4-12 and 6-20, Exhibit 4.5).

In addition to the M-DBP, three Small Business Advocacy Review Panels (SBARs) were created pursuant to § 244 of the Small Business Regulatory Enforcement and Flexibility Act of 1974 (SBREFA 1996). Each panel included multiple stakeholders and nongovernmental experts (USEPA 2018, 2019). According to Raucher and Cromwell (2004), USEPA's *ex ante* compliance cost estimates were not controversial largely because of stakeholder participation. However, because USEPA predetermined that the MCLs for total trihalomethanes, haloacetic acids, and bromate that were proposed prior to SDWA 1996 would not be changed (USEPA 2009b, Sec. 2.1), the M-DBP and SBARs did not examine the rules' economic feasibility.

The two DBP rules were intended to reduce (primarily) bladder cancer risks assumed to result from exposure to disinfectants used to reduce certain pathogens. In 1996, bladder cancer incidence was 36.7 per 100,000 for men and 20.8 per 100,000 for women. NCI's SEER 9 data indicate that rates rose a statistically significant 0.2% from 1986-2007 and declined a statistically significant 1.1% from 2007-2015 (NCI 2018a). Relative to the approximately 80,000 new bladder cancer cases per year reported (NCI 2018b), the number of bladder cancer cases attributed by USEPA to drinking water disinfectants has been uncertain (USEPA 1998a, p. 69434) or small (USEPA 1998d, p. 4-21 [0–9,300]; 2005b, Exhibit 6.20 [275–874]). USEPA acknowledged that no causal relationship had been established (USEPA 2005b, p. 6-65).

The three surface water treatment rules and the ground water rule were intended to control cryptosporidiosis and giardiasis from parasites that are resistant to inactivation by disinfection. Cryptosporidiosis and giardiasis result from the ingestion of sufficient *Cryptosporidium* oocysts or *Giardia* trophozoites through the consumption of fecally

contaminated food or water, or person-to-person or animal-to-person transmission (Fayer et al. 2000). Most cases go undiagnosed, few who are infected seek medical attention, hospitalization is rare. Mortality from cryptosporidiosis is rarer still and highly concentrated among immunocompromised subpopulations; mortality from giardiasis is virtually nonexistent (Painter et al. 2015a; Painter et al. 2015b; CDC 2017b, c). Both parasites are endemic, but infections are associated with warm weather use of recreational water, which has been estimated to be responsible for 37% of all cases of cryptosporidiosis (CDC 2017a, p. 14).

In 1999, incidence of cryptosporidiosis across the continental U.S. ranged from 0.1 to 7.2 cases per 100,000, with a mean of 1.0 case per 100,000 (Hlavsa et al. 2005). By 2011, the mean and range had grown to 3.0 and 17.7 per 100,000, respectively (Painter et al. 2015b). FDA's FoodNet (a nonrepresentative sample of 10 States comprising about 15% of the U.S. population) reported no statistically significant change in foodborne cryptosporidiosis incidence from 2006-13 (Gilliss et al. 2013; Crim et al. 2014). Underreporting is believed to be widespread because cryptosporidiosis and giardiasis are typically mild gastrointestinal infections, etiologic confirmation requires laboratory analysis, and confirmation of the underlying pathogen has no value added for guiding treatment decisions. Hospitalization is infrequent and mortality is rare, especially since the development of retroviral drugs to treat AIDS. From 2000-17, the average annual numbers of nationwide cryptosporidiosis and giardiasis cases were 7,274 and 16,747, respectively (Centers for Disease Control and Prevention 2018a). The average annual numbers of nationwide mortalities were 1.74 and 0.42, respectively (Centers for Disease Control and Prevention 2018b).

Finally, two rules in this suite were intended to control potential viral and bacterial infections attributed to ground water used as drinking water. Like the other regulations, the causal nexus between drinking water and illness is unclear and estimated benefits are small.

USEPA risk assessment practices would have materially adverse effects on economic feasibility determinations in the same way they affect determinations that benefits justify costs. This is further discussed in (h) below.

(a) Stage 1 Disinfectants and Disinfection Byproducts Rule (Stage 1 DBPR)

In this rulemaking, USEPA revised NPDWRs for disinfectants and DBPs, which included maximum residual disinfectant levels (MRDLs), maximum contaminant levels (MCLs), and treatment techniques. Community Water Systems (CWSs) and Non-Transient Non-Community Water Systems (NTNCWSs) were generally covered, and some provisions also applied to Transient Non-Community Water Systems (TNCWSs). MRDLs and MCLs were based on technological feasibility (USEPA 1998a); cost played no role. Nonetheless, the Agency asserted that the benefits of the rule justified the costs, thus making this rule potentially susceptible to economic feasibility analysis.

USEPA estimated aggregate expenditures for treatment and monitoring at $701 million ($1998, annualized at 7%) including capital, operations and maintenance, and monitoring (USEPA 1998a, Table IV-6). Expenditures for DBP control depended on technology and system size. The range across technologies for surface water systems was as great as 162x, and the range across system sizes was as great as 608x. Similar variability was reported for ground water systems (USEPA 1998a, Tables IV-4 and IV-5). Raucher and Cromwell (2004) estimated that 40% of expenditures would be borne by systems serving fewer than 10,000 persons.

Nearly 12,000 such systems were expected to be required to modify their treatment trains and 62,000 small systems were expected to expend significant new resources to collect over 1 million new monitoring samples (USEPA 1998d, Exhibits 5.8 and 5.9). An estimated 96% of CWSs and NTNCWSs serve fewer than 10,000 persons, and 22% of them were estimated to bear additional costs. About 8,200 small groundwater systems and 3,600 surface water systems were estimated to expend about $1 billion and $240 million in capital improvements, respectively, for modified treatment.

USEPA analyzed small-system cash flow and viability to gain insight on the risk of financial failure, a proxy for system affordability. USEPA acknowledges that small systems would have preferred that the Agency promulgate different standards for large and small systems to avoid small-system nonviability and forced consolidation, but the Agency did not analyze these options and there is no evidence that it gave them serious consideration (USEPA 1998d, pp. 5-23 to 5-25). The Agency's economic analysis does not consider forced consolidation with larger systems to be a social cost because consolidation is a longstanding Agency goal (USEPA 1998d, pp. 5-19 to 5-23).

The Agency acknowledged that benefits were "difficult to quantify because of the uncertainty associated with risks from exposure to DBPs (and the resultant reductions in risk due to the decreased exposure from DBPs)" (USEPA 1998a, p. 69434). These uncertainties include a nine-fold range in the Agency's estimated number of bladder cancer cases caused by DBPs, plus an acknowledgement that the true number of cases could be zero (USEPA 1998a, Table IV-8; 1998d, Exhibit 6.3c). Estimated aggregate annual benefits ranged from $0 to $4 billion, with the upper-bound implying a 24% reduction in new bladder cancer cases resulting from a 24% reduction in exposure (USEPA 1998d, p. 6-1). No cessation lag appears to have been applied.

USEPA's net benefit estimates were derived by subtracting a range of expenditures ($500–$900 million) from the $0–4 billion range of opportunity cost-based health benefits. The Agency described these ranges as "overlapping," thus providing a "substantial basis" for the rule (USEPA 1998d, p. 6-2). Net benefits depended on the number of annual bladder cancer cases prevented, and USEPA acknowledged that taking no action is a superior choice if the number of cases prevented is likely to be below 1,000–2,500 per year (USEPA 1998d, Exhibit 6.2).

In the end, USEPA concluded that "there is a reasonable likelihood that the benefits will exceed the costs" (USEPA 1998a, p. 69434), and that the rule was "needed for protection of public health from exposure to DBPs" due to the extent of potential exposure, evidence of DBP carcinogenicity and reproductive or developmental harm in laboratory animals at high doses, and weak associations in some epidemiological studies (USEPA 1998c, p. 5-15). Thus, the new standards were grounded in precaution with respect to *potential* health risks without regard for *actual* opportunity costs.

The extent to which prior adoption of the economic feasibility principle would have enhanced efficiency or reduced inequity cannot be determined based on the information disclosed, however. The selected alternative was the only one examined in the economic analysis (USEPA 1998c, p. 6-3) and only nationwide estimates of benefits and costs were reported, without disaggregation by system size, region, or other key factors. However, if USEPA had adopted the economic feasibility principle *and* followed the risk- and economic analysis requirements in SDWA 1996, the Agency would have either adopted less stringent standards or declined to modify existing standards.

(b) Interim Enhanced Surface Water Treatment Rule (IESWTR)

The IESWTR (USEPA 1998b) was a response to the 1993 cryptosporidiosis outbreak in Milwaukee, which was attributed to a sudden increase in *Cryptosporidium* in source water at the intake for the treatment plant on the city's south side combined with treatment failure (Fox and Lytle 1996). Turbidity (a measure of suspended particulates, and thus a proxy for *Cryptosporidium*) in finished water increased from about 0.4 nephelometric turbidity unit (NTU) to a peak of 2.7 NTUs at the height of the crisis (Fox and Lytle 1996, p. 92 and Figure 2). Turbidity is an imperfect indicator of *Cryptosporidium*; other water systems utilizing Lake Michigan also experienced increases in turbidity but did not have a cryptosporidiosis outbreak (Fox and Lytle 1996, p. 94 and Table 1).

Using a RDD telephone survey, Mac Kenzie et al. (1994) estimated more than 400,000 persons were "affected" by the outbreak; cryptosporidiosis was clinically confirmed in more than 600 people. Cryptosporidiosis can be "unrelenting and fatal" in those who are immunocompromised (p. 161). Hoxie et al. (1997) report that AIDS was the underlying cause of death recorded in 46 (85%) of the 54 cryptosporidiosis-associated fatalities. AIDS mortality declined for about 18 months thereafter, then resumed its previous upward trend. Hlavsa et al. (2005) reported that cryptosporidiosis-associated fatalities among HIV-infected subpopulations decreased substantially after the Milwaukee outbreak due to the contemporaneous introduction of effective antiretroviral therapy.

IESWTR required water systems serving at least 10,000 persons that relied on surface water sources to use a treatment technique in lieu of achieving an MCL, and meet other regulatory controls (USEPA 1998b). Increasing the stringency of turbidity control was assumed to reduce *Cryptosporidium* and *Giardia* in finished water. The Agency asserted that the

regulation's benefits justified the costs (USEPA 1998b, pp. 69499, 69509), so the rule is potentially susceptible to economic feasibility analysis.

For benefit estimation, USEPA relied on health risk-precautionary risk assessment methods and data from the Milwaukee outbreak (USEPA 1998c, § 4.2); cost of illness (COI) valuations derived from previous survey research on giardiasis (Harrington et al. 1985); and assumptions about the effectiveness of turbidity reduction in removing *Cryptosporidium* oocysts (USEPA 1998c, § 4.2.4).

The Agency's mid-level estimates of regulatory effectiveness were annual reductions of 432,000 illnesses and 60 deaths (USEPA 1998c, Exhibits 4.12 and 4.16). However, the number of illnesses and deaths reported from all pathways in 1998 was 3,111 and zero, respectively (Centers for Disease Control and Prevention 2018a). Fewer than two deaths per year were reported for 1999-2017 (Centers for Disease Control and Prevention 2018b). USEPA's risk model did not account for reductions in mortality among immunocompromised individuals resulting from the inexpensive implementation of certain best practices (e.g., bottled water) or antiretroviral therapy for AIDS. Even before the ISESTWR was promulgated, it was suspected that the underlying cause of the Milwaukee outbreak was a failure in sewage treatment upstream of the drinking water intake (Peng et al. 1997).

USEPA reported estimated aggregate annualized expenditures of $596 million for disinfection and $189 million for turbidity reduction (both $1998, 7%), the latter expenditure substantially varying by system size (USEPA 1998c, Tables V.1 and V.2). USEPA's affordability doctrine figured implicitly in the Agency's household-level expenditures, which were $12 or less per year for 92% of households; $12–60 per year for 7% of households; and "approximately $100 per year" for the remaining 1% of households (USEPA 1998c, Figure V.1;

1998c, p. 5-25). No further breakdown by system size was provided. The economic analysis included a brief, qualitative opportunity-cost justification of the lower-end of these estimated household-level expenditures (USEPA 1998c, ES-6). Aggregate expenditure estimates were reported to span ± 30% (USEPA 1998b, p. 69437).

The Agency reported opportunity-cost based benefit estimates that varied depending on the assumed number of illnesses and mortalities avoided, and the level of *Cryptosporidium* removal (USEPA 1998c, Table V.4). Regulatory effectiveness was acknowledged to be highly uncertain. The number of illnesses estimated by USEPA to be prevented ranged from zero to 1.029 million, and the estimated number of fatalities prevented ranged from zero to 129. Illnesses were valued at $2,000/each using COI methods. Premature mortalities, concentrated among immunocompromised persons, were valued using the then-applicable VSL (USEPA 1998c, ES-4).

Several potential unquantified benefits also were mentioned (USEPA 1998b, p. 69501). Unquantified costs were not discussed and opportunity costs were not estimated, however. Thus, to conclude that benefits justified costs, USEPA relied on a range of *opportunity-cost derived* benefit estimates ($0–$2.7 billion) encompassing a point estimate of annualized *compliance* expenditures ($633 million) (USEPA 1998b, p. 64509; 1998c, pp. ES-7). The relative probabilities of these estimates were not taken into account.

Like in the case of the Stage 1 DBPR, savings from prior adoption of the economic feasibility principle cannot be determined based on the information disclosed. Moreover, USEPA's decision appears to have been driven by its internal policy preferences for precaution with respect to uncertain risks (but not certain costs) and quantity-based equity. Finally, as discussed in (h) below, USEPA's estimated numbers of illnesses and deaths from

cryptosporidiosis prevented significantly exceed estimates of baseline illnesses and deaths reported by CDC.

(c) Filter Backwash Recycling Rule (FBRR)

During the consideration of SDWA, Congress became aware of concerns that the routine recycling of filter backwash water could inadvertently result in recontamination of finished water. SDWA 1996 § 1412(b)(14) directed USEPA to manage this risk, which it did via the FBRR (USEPA 2001b). Annualized costs were estimated at about $7 million ($2000, 7%) (Exhibit 6-5). Raucher and Cromwell (2004) note that post-promulgation cost estimates were considerably higher. Potential benefits were described qualitatively, so the Agency's determination that benefits justified costs was qualitative as well (USEPA 2001d, p. 31096; 2000b, p. 8-2). The potential effects of having relied on the economic feasibility principle cannot be estimated without quantitative benefit estimates.

(d) Long Term 1 Enhanced Surface Water Treatment Rule (LT1ESWTR)

This regulation extended the IESWTR to systems serving fewer than 10,000 persons (USEPA 2002b). The Agency determined that the benefits of the regulation justified the costs (USEPA 2002b, p. 1827), making this rule potentially susceptible to economic feasibility analysis.

USEPA estimated combined annualized system and State compliance expenditures at $44.8 million ($1999, 7%) (USEPA 2002b. p. 1822). Key expenditures (e.g., for land acquisition) and opportunity costs generally (e.g., the value of owned land newly committed to treatment) were not estimated. System-level expenditures were assumed to be passed through to households. Annual expenditures per household were estimated at $6.24 (mean), $15.00 (90th

percentile), $15–$120 (91st to 98th percentile), and over $120 for the 99th percentile. Expenditures were estimated to exceed $240 per year for 5,600 households. USEPA reported having estimated benefits for three assumed risk levels and two baseline assumption for pre-regulation cryptosporidium removal. Illness cases were valued at $796–$1,411 per case based on the cost of treating giardiasis (an illness of longer duration), and mortalities were valued at the then-applicable VSL.

USEPA's reported net benefit calculations subtract a point estimate for annualized expenditures ($44.8 million) from a range of opportunity-cost based benefits ($18.9–$90.9 million) (both $1999). The extent to which adoption of the economic feasibility principle would have reduced costs or increased net benefits cannot be determined from the information made public because key analytic documents (USEPA 2000g, 2001a) referenced in the preamble to the final rule were not included in the docket and are not electronically available to the public despite an aggressive Google search. Finally, as noted in subsection A1(h), USEPA's estimated numbers of illnesses and deaths from cryptosporidiosis prevented significantly exceed the numbers of illnesses and deaths from all pathways reported by CDC.

(e) Stage 2 Disinfectants and Disinfection Byproducts Rule (Stage 2 DBPR)

This regulation expanded upon the Stage 1 DBPR, targeting systems in compliance but with peak concentrations greater than Stage 1 MCLs (USEPA 2006d). Aggregate annualized system expenditures were estimated at $76.8 million ($2003, 7%) for nine system-size categories (USEPA 2006d, p. 445). Unquantified costs were not identified (USEPA 2005b, p. 7-47). Increases in mean household-level expenditures were estimated at $4.58 or less, depending on system size (USEPA 2006d, Table VI.E-1), and appear to be consistent with the affordability doctrine. Opportunity costs to systems or their customers were not estimated.

Annualized benefit estimates consist of the value of "reasonable estimate[s] of the range of potential [bladder cancer] risks" from peak DBP exposure, acknowledging that "the existing epidemiological evidence has not conclusively established causality between DBP exposure and any health risk endpoints." USEPA acknowledged that "potential risks may be as low as zero" but nevertheless assumed causality. The Agency attributed 2–17% of baseline bladder cancer cases to DBPs (USEPA 2006d, p. 444). Depending on the risk model used (all of which excluded zero risk), the estimated number of cancers prevented ranged from 61 to 610 (USEPA 2006c, Table VI.C-1).

Unquantified benefits are discussed, notably reproductive and developmental risks derived from "possible associations" reported in certain epidemiological studies described as having "mixed" results (USEPA 2006d, p. 391). USEPA further asserted that the rule might provide these additional benefits because "a weight of evidence evaluation of the health effects data suggests a potential association" (USEPA 2005b, p. ES-11). USEPA's net benefit calculation was positive (Table VI.F-1), but as before was obtained by subtracting *virtually certain expenditures* from the value of *potential benefits*. The number of cancers that must be prevented to cover annualized opportunity costs cannot be calculated because opportunity costs were not estimated.

Adoption of the economic feasibility principle may have yielded a substantially different decision because risk and benefit per person are assumed to be linear while expenditures per person vary greatly by broad system size category and source water, as shown in Table A1. Mean annualized expenditures per household range from $2.83 to $49.69 per year, a span of more than 17x, depending on system size. At the 90th percentile, estimated household

expenditures range from $6.98 to $173.53, a factor of 25x. Detailed results by system size are reported in an appendix to USEPA (2005b) that is not publicly available, however.

USEPA acknowledged that the scope of potential exposure (> 260 million persons) "played a significant role" in its decision to promulgate the Stage 2 DBPR, not the strength of evidence for risk. USEPA also described the rule as necessary to ensure "more consistent, equitable protection from DBPs" (USEPA 2006d, p. 391), thus indicating that achieving quantity-based equity was a significant consideration.

Table A1: Stage 2 DBPR: Annualized Expenditure Increases per Household, Breakeven Annualized Cancer Cases per household, and Breakeven Minimum Households per Cancer Case Avoided

Annualized Expenditure/ Household	System Category	Breakeven Minimum Annualized Cancer Cases per Household	Breakeven Minimum Households per Cancer Case Avoided
AVERAGE			
$ 2.83	SW > 10k	2.53×10^{-6}	395,505
$ 49.69	GW < 10k	4.44×10^{-5}	22,525
90th PERCENTILE			
$ 6.98	SW > 10k	6.24×10^{-6}	160,355
$ 173.53	SW < 10k	1.55×10^{-4}	6,450
Derived from USEPA (2005a). Value of prevented fatal and nonfatal bladder cancer = $2.028 million and $0.8 million, respectively. Proportion of bladder cancers assumed fatal = 26%.			

(f) Long Term 2 Enhanced Surface Water Treatment Rule (LT2ESWTR)

This regulation expanded regulatory requirements for the treatment of surface waters used for drinking water for the control of *Cryptosporidium* and other microbial pathogens resistant to disinfection (USEPA 2006c). The rule targeted systems that rely on surface waters with unusually high *Cryptosporidium* occurrence (not concentration) rates; subjected to

treatment requirements systems that previously were not required to filter; and required existing finished water storage reservoirs to be covered. USEPA determined that benefits justified the costs (USEPA 2006c, p. 748), thus making the rule potentially susceptible to economic feasibility analysis.

USEPA examined three alternatives besides the Agency's preferred alternative (A3) (USEPA 2005a, Chapter 3), which the Agency described as "the most cost-effective and [] deliver[ing] the best value" (USEPA 2005a, p. 3-3). These alternatives vary principally by monitoring frequency, with resulting treatment expenditures determined by monitoring results. No formal alternatives were examined with respect to provisions that were agreed upon through the Stage 2 M-DBP Reg Neg. Compliance with the LT1ESWTR was assumed in the analytic baseline.

Selected point estimates for aggregate compliance expenditures (not opportunity cost) are provided in Table A2, for three different data sets that the Agency said were "equally likely to represent the true distribution of *Cryptosporidium* in source waters for all systems" (USEPA 2005a, ES-8). Estimates of annualized household-level expenditures are not clearly inconsistent with the affordability doctrine, though expenditures for households at the upper end might not have been affordable (USEPA 2005a, Exhibit ES.8-3). Unquantified costs were briefly discussed but not otherwise accounted for (USEPA 2005a, pp. 6-33 to 6-34).

As shown in Table A3, quantified benefits consist of estimated averted illnesses and deaths, which were valued at $844 and $7.4 million per case ($2003), respectively. Depending on the data set, estimated annual cases (deaths) averted ranged from 146k to 500k (38 to 130), a range of 3.4x (USEPA 2005a, Exhibits 5.16 and 5.17). Risk was estimated using a simulation model (USEPA 2005a, § 5.2). Certain forms of variability and uncertainty (e.g., exposure via

drinking water) were captured by stochastic distributions. Uncertainty about morbidity subsequent to exposure and mortality subsequent to infection are modeled as assumptions. Thus, estimated benefits critically depend on the accuracy of these embedded assumptions. As noted in subsection A1(h), all model outputs of illnesses and deaths prevented significantly exceed the numbers of cases and deaths from all pathways reported by CDC.

Morbidity and mortality were valued using cost of illness (COI) methods and WTP, respectively, under multiple scenarios reflecting substantial uncertainty about hazard, dose-response, exposure, risk model structure, and limited data (USEPA 2005a, Chapter 5). The estimated numbers of illnesses averted were reported with up to seven significant figures, ranging from 363,328 to 1,501,445, depending on risk model and regulatory alternative/data set combination (USEPA 2005a, Exhibit 5.5). The Agency modeled illness risk from *Cryptosporidiosis* exposure as a triangular distribution (30%, 50%, 70%), and further assumed that the probability of mortality given illness was 24.07 per 100,000 illnesses among AIDS patients served by unfiltered systems, 14.56 per 100,000 illnesses for AIDS patients served by filtered systems, and 1.98 deaths per 100,000 for all others, with no uncertainty (USEPA 2005a, p. 5-22).

Risk and benefit estimates were disaggregated to reflect variability by system type and size, baseline filtration, and *Cryptosporidium* occurrence. The Agency also provided an extensive discussion of potential unquantified benefits. Mean net benefits were calculated by subtracting expenditures from opportunity-based benefit estimates, and were positive for most scenarios using both conventional and "enhanced" COI methods (USEPA 2005a, Exhibit 8.12a, b). The reported dominant uniform regulatory alternative depends on the choice of data set and discount rate (3% and 7% were examined), but uncertainty in the illness risk component of the

model clearly dominates both data set choice and discount rate. Small changes in the parameters of the triangular distribution would swamp estimated reductions. Moreover, any validity that might have attached to USEPA's results was vitiated by the assumption that qualitative benefits were five times as great as quantified benefits (USEPA 2006c, Table VI.C-1; 2005a, p. 8-21).

The likely effect of the economic feasibility principle is difficult to ascertain because the Agency's analysis was structured in ways that made estimated net benefits more likely. In addition, detailed estimates are needed by system size to determine where to draw the line between covered and exempt systems. USEPA analyzed benefits and costs by system-size categories but only reported aggregate estimates in the main analysis. Twenty-one appendices to the economic analysis contained, among other things, the Agency's analysis of small-system impacts. They are not publicly available.

Table A2: USEPA-estimated Aggregate Annualized Expenditures for LT2ESWTR

Data Set [a]	Total Annualized Costs $M (3%)			Total Annualized Costs $M (7%)		
		90% Confidence Bound [b]			90% Confidence Bound [b]	
	Mean	Lower (5th %ile)[b]	Upper (95th %ile)[b]	Mean	Lower (5th %ile)[b]	Upper (95th %ile)[b]
ICR	$133	$111	$160	$150	$125	$181
ICRSSL	93	72	112	107	83	129
ICRSSM	106	86	126	121	99	144

Source: (USEPA 2005a, Exhibit 8.5]).
[a] ICR = 1996 Information Collection Rule; ICRSSL = ICR Supplemental Surveys (Large Systems); ICRSSM = ICR Supplemental Surveys (Medium-Size Systems).
[b] Meaning of 5th and 95th percentiles with 90% confidence bounds is not explained.

Table A3: LT2ESWTR: USEPA-estimated Annual Illnesses and Deaths Avoided

Data Set [a]	Annual Illnesses Avoided			Annual Deaths Avoided		
		90% Confidence Bound [b]			90% Confidence Bound [b]	
	Mean	Lower (5th %ile)[b]	Upper (95th ile)[b]	Mean	Lower (5th %ile)[b]	Upper (95th %ile)[b]
Annual Total after Full implementation						
ICR	964,360	149,241	2,277,367	207	34	468
ICRSSL	230,730	38,281	521,925	52	9	113
ICRSSM	455,170	72,128	1,112,374	100	17	230
Annual Average over 25 Years						
ICR	712,732	109,486	1,685,176	154	25	348
ICRSSL	170,977	28,314	392,979	39	7	85
ICRSSM	336,652	52,763	826,004	74	12	172

Source: (USEPA 2005a, Exhibit 8.3]).
[a] ICR = 1996 Information Collection Rule; ICRSSL = ICR Supplemental Surveys (Large Systems); ICRSSM = ICR Supplemental Surveys (Medium-Size Systems).
[b] Meaning of 5th and 95th percentiles with 90% confidence bounds is not explained.

(g) Ground Water Rule (GWR)

This regulation added an additional layer of treatment requirements for ground water systems believed by USEPA to be susceptible to fecal contamination "because such contamination is the likely source of viral and bacterial pathogens in drinking water supplies" (USEPA 2006b, p. 65576). USEPA determined that the risk-targeting strategy was cost-effective (USEPA 2006a, Chapter 8) and that the benefits justified the costs (USEPA 2006b, p. 65637). Thus, this rule is potentially susceptible to economic feasibility analysis.

The regulation's "risk-targeting strategy" included triennial or quintennial sanitary surveys with survey-triggered monitoring requirements, mandatory corrective action by treatment, and compliance monitoring. Costs were estimated conventionally based on paperwork, recordkeeping, and engineering expenditures, not opportunity costs. Benefits were based on potential risks and estimated using both "traditional" and "enhanced" COI methods. "Enhanced" methods included the value of lost nonmarket work time based on opportunity costs, the value of lost leisure time, and the value of lost productivity (USEPA 2006a).

Figure Aa shows USEPA estimates of benefits and costs of the selected regulatory alternative for two benefit estimation methods ("enhanced" and "traditional" COI) and two discount rates (3% and 7%) (USEPA 2006b, Table VII-1). Aggregate net benefits at the mean, 5th percentile, and 95th percentile were negative regardless of dataset, discount rate, and COI methodology.

The determination that benefits justified costs depended on substantial unquantified co-benefits and the virtual absence of unquantified costs. Unquantified co-benefits were assumed to be at least four times greater than quantified benefits (USEPA 2006a, § 5.4.3.2). Unquantified costs were assumed to be minor (USEPA 2006a, § 6.6). Three alternatives were analyzed, and

aggregate net benefits also were negative for each. Alternative 1 had the smallest negative net benefits (–$11.7 million $2003 at 3%; –$12.4 million $2003 at 7%) (USEPA 2006a, Exhibit 8.10a).

Net benefits objectively estimated would have been more severely negative. As noted in subsection A1(h), USEPA's estimates of illnesses and deaths prevented were significantly greater than the baseline numbers of cases and deaths from all pathways reported by CDC. USEPA's institutional reliance on risk assessment methods that yield upwardly biased estimates of risk (USEPA Office of the Science Advisor 2004) result in upwardly biased estimates of benefits.